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Mapping eutrophication risk from climate change: Future phosphorus concentrations in English rivers

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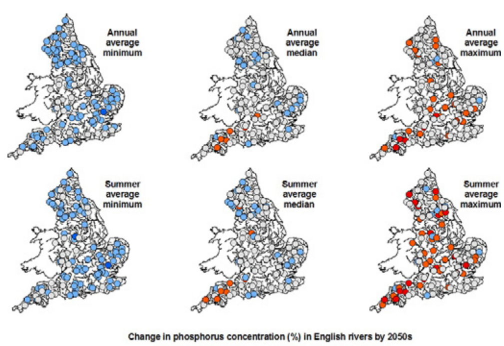
HIGHLIGHTS

- Climate change to increase phosphorus concentrations in rivers by 2050
- Small but inconsistent increase in phosphorous in English rivers
- Improved sewage treatment inadequate to meet existing phosphorus standards

GRAPHICAL ABSTRACT



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ABSTRACT

Climate change is expected to increase eutrophication risk in rivers yet few studies identify the timescale or spatial extent of such impacts. Phosphorus concentration, considered the primary driver of eutrophication risk in English rivers, may increase through reduced dilution particularly if river flows are lower in summer. Detailed models can indicate change in catchment phosphorus concentrations but targeted support for mitigation measures requires a national scale evaluation of risk.

In this study, a load apportionment model is used to describe the current relationship between flow and total reactive phosphorus (TRP) at 115 river sites across England. These relationships are used to estimate TRP concentrations for the 2050s under 11 climate change driven scenarios of future river flows and under scenarios of both current and higher levels of sewage treatment.

National maps of change indicate a small but inconsistent increase in annual average TRP concentrations with a greater change in summer. Reducing the TRP concentration of final sewage effluent to 0.5 mg/L P for all upstream sewage treatment works was inadequate to meet existing P standards required through the EU Water Framework Directive, indicating that more needs to be done, including efforts to reduce diffuse pollution.

1. Introduction

Eutrophication is seen as one of the most serious problems facing river ecology worldwide (Carpenter et al., 1998; Mainstone and Parr, 2002). The nutrient enrichment of rivers from point and diffuse sources

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can reduce ecological status, often resulting in excessive plant and algal growth, ecological regime shifts and problems associated with low oxygen concentrations such as fish kills (Hilton et al., 2006). Eutrophication also has associated economic costs due to lost amenity value, impacts on property prices and tourism, and increased water treatment costs (Pretty et al., 2003). Within the European Union, the Water Framework Directive (WFD) requires nation states to ensure the health of freshwater ecosystems and to avoid the ecological problems associated with eutrophication (Hutchins, 2012). Eutrophication is a future risk identified in the UK Climate Change Risk Assessment (DEFRA, 2012a) and is an area where evidence and planning needs to be improved (DEFRA, 2012b). The full extent of existing eutrophication is not fully known or understood (Environment Agency, 2012). Phosphorus (P) is seen as the major contributor to eutrophication in fresh waters, with 45% of rivers and 76% of lakes in England currently failing their P concentration standard for good ecological status (Environment Agency, 2012). Future climate change is expected to impact on UK river flows, and this in turn will influence future P concentrations (Johnson et al., 2009). Understanding how P concentrations may change in the future is a key stage in developing a greater understanding of future eutrophication risk and for the design and targeting of management solutions.

Despite extensive research into the processes and impacts of eutrophication and water quality, climate change impacts have been less well studied (Whitehead et al., 2009). Climate change may increase the risk of higher pollution concentrations and excessive algal growth in water bodies due to projected reduction in summer flows and higher water temperatures (e.g. Whitehead et al., 2009; Jeppesen et al., 2010; Moss et al., 2011; Moss, 2012; Whitehead and Crossman, 2012; Johnson et al., 2009; Hutchins et al., 2016). Changing temperature and hydrology will affect multiple levels of biological organisation and interact with other freshwater stressors (Woodward et al., 2013). Fewer studies have considered these changes in rivers where interactions with hydrology make for a different and less tractable set of controls than in lakes (Hutchins, 2012). Eutrophication is a complex phenomenon representing multiple interlinked processes and a range of field and modelling approaches have been used to assess individual components of the system, often giving site or catchment-specific insights (Whitehead et al., 2009). Generalised understanding that has emerged suggests that climate change may increase eutrophication primarily through two routes:

1. Higher temperatures increase the rate of biological and chemical processes (in particular increasing algal growth rates and nutrient cycling).
2. Decreased summer flows will reduce dilution of constant nutrient inputs from sewage treatment works effluent (Bowes et al., 2008), and increase residence times of water, which could lead to increased phytoplankton biomass (Bowes et al., 2012a).

As an important first step to understanding the impact of climate change on future eutrophication risk, we investigate how phosphorus concentrations are affected by predicted flows and improvements in STW. In this paper we focus on P as it is the main limiting factor to algal growth in English rivers. Nitrogen concentrations may be limiting in occasional cases but in some catchment/region-specific studies nitrogen is far in excess (e.g. Bowes et al., 2012a, b) and for eutrophication is less limiting than phosphorus (e.g. Hutchins, 2012). Academic datasets show that in most rivers in England, such as the CEH lowland river water quality dataset (Neal et al., 2012), inorganic forms of phosphorus and nitrogen dominate the nutrient loadings. Organic nutrients will only become an important consideration when inorganic nutrient concentrations reach potentially-limiting concentrations. Eutrophication risk is complex and depends on a range of factors. For example, levels of phytoplankton biomass may be subject to biological control, notably by zooplankton grazing (Schol et al., 2002; Descy et al., 2003) and also by algicidal bacteria and fungal parasites. The removal of dams causes localised changes to the hydraulic depth and flow velocity of the river (Cisowska and Hutchins, 2016) and the resulting changes to nitrogen cycling has been estimated. Such changes are also likely to affect phosphorus and sediment fluxes. Projected rises in population will increase the loading of phosphorus to rivers. By 2039 the UK population is set to increase by 15% from present day levels (ONS, 2015). Although other pressures such as land use change and population growth will also affect the risk of eutrophication and future phosphorus levels, understanding the impact of river flows is a critical first step to developing an understanding of future patterns of change and risk. Furthermore, given the complexities of nutrient cycling in the fluvial system, a range of other climate change impacts may influence both phosphorus concentrations and eutrophication risk. For example, sediment and nutrient delivery may increase during high rainfall events; the frequency of which may increase under climate change (see, for example Ockenden et al., 2016). It has also been demonstrated that threshold conditions in phosphorus, flow, temperature and sunlight need to be met before algal blooms can develop (Bowes et al., 2016). Future estimates of phosphorus concentrations based on river flows under the influence of climate change are an essential precursor to developing a capability to estimate future eutrophication risk to guide national adaptation measures.

The aim of this study is to estimate change in river phosphorus concentrations under different future climate change scenarios; specifically:

1. How P concentrations change under different scenarios of climate driven river flow
2. What do climate changes mean for WFD objectives and P standards?
3. Do plausible changes to waste water treatment for P help mitigate climate effects?

We use a Load Apportionment Model (LAM) to establish present day relationships between P concentration and flow for sites across England, and use this to estimate P concentrations to the end of the 21st century under 11 future flow scenarios and a waste water treatment improvement scenario.

2. Methods

Our approach uses river monitoring data from multiple sites across England to establish the present day relationship between P concentration and daily mean flow. This is then used to predict changes in P concentration due to projected changes in river flow due to future climate change. Source apportionment can identify the contribution by different sectors to water pollution and has been used to assess the change in P concentrations with change in river flow as a result of abstraction management (e.g. Bowes et al., 2008, 2010; European Environment Agency, 2005). This approach has not previously been used to look at climate change beyond a very limited area (e.g. Atkins, 2014).

2.1. Load apportionment modelling

The Load Apportionment Model (LAM) (e.g. Bowes et al., 2008) offers a simple yet relatively robust method for estimating the relative loads of point and diffuse inputs to a river, founded entirely on routinely-collected concentration and flow data. Other catchment data, such as land use, human and livestock densities, fertiliser application rates etc. are not required, and so the LAM approach is ideally suited for application to multiple catchments on a national scale. The approach is based on the observation that rivers that receive the majority of their P inputs from Sewage Treatment Works (STW) always have their highest P concentrations at lowest flows, and this rapidly decreases with increasing flows. This is because the daily STW effluent inputs to rivers are relatively constant throughout the year, and therefore the dilution of this constant input within the river is at its lowest when flow is at its minimum. As the river flow increases due to rainfall, these dominant STW inputs will be diluted, and hence the P concentration/flow relationship produces a dilution curve. Conversely, rivers that receive no

STW point inputs will not exhibit this dilution relationship. Rivers dominated by diffuse, rain-related inputs will exhibit increasing P concentrations/loads with increasing river flow. Dilution curves can also be observed if groundwater P signals are high (although most groundwater-dominated rivers in England have calcareous geology that will largely precipitate out the dissolved P), and similarly rising P concentrations with flow can occur due to sewer overflows. These are implicit in LAM development, which retains a simple approach applicable at a national scale. In addition, the sites considered here tend to be dominated by surface water contributions (see BFI in Table 1).

The model produces a line of best fit to the empirical data by applying a constant (point source) STW component (consisting of a simple dilution curve) and a rain-related (diffuse) component (consisting of 2 parameters, describing the quantity of diffuse phosphorus inputs and how this input responds to increasing river flow (a gradient component)). A full description of how the model operates is available elsewhere (Bowes et al., 2008, 2009a, b). In brief, the phosphorus concentration, C_p (mg m^{-3}) at the monitoring point can be expressed as:

$$C_p = A.Q^{-1} + C.Q^{D-1} \tag{1}$$

where Q (m^3 s^{-1}) is the volumetric flow rate of the river, and *A*, *C* and *D* are load coefficients to be determined empirically. The $A.Q^{-1}$ term is the nutrient concentration originating from 'constant' (i.e. non flow-related) sources. The *A* parameter equates to point source sewage effluent. The $C.Q^{D-1}$ term in Eq. (1) is the nutrient concentration originating from rainfall and flow-related sources, and will largely equate with diffuse source inputs. The model solution is the sum of the constant source contribution (derived from the *A* load coefficient) added to the rain-related source contribution (derived from the *C* and *D* terms).

2.2. Flow data

Paired data sets were used to establish current relationships between flow and P concentration. These were then applied to locations where there were future projections of river flows. For this we used projections known as Future Flows Hydrology (FFH) which were developed as part of the Future Flows and Groundwater Levels project[1] (Prudhomme et al., 2012a, 2013). FFH provides daily river flow and groundwater level transient projections for 282 river catchments and 24 boreholes across the UK for 1951–2098. These were derived from Future Flows Climate (FFC) (Prudhomme et al., 2012a) using a range of hydrological models. FFC is an ensemble of 1 km gridded transient projections of precipitation and potential evapotranspiration based on 11 variants of the Hadley Centre Regional Climate Model (HadRM3-PPE). FFH provides a nationally consistent ensemble of 11, equally likely, plausible realisations of the river flow and groundwater level regime under a future world scenario that has high economic growth, is integrated and uses a balance of energy sources (SRES A1B emission scenario, see Nakicenovic et al., 2000). Considering all ensemble members together accounts for some of the uncertainty around future climate change. This study investigated the 150 flow-gauged river monitoring sites throughout England where time series of future flows hydrology (FFH) have been derived for the period 1951 to 2098 (Prudhomme et al., 2012a, 2013).

2.3. Phosphorus and flow data

Of the 150 Future Flows Hydrology sites, 115 sites (Fig. 1 and Table 1) were selected because they had paired historic and current observations of flow and phosphorus concentration data and no subsequent problems for model fit. Total reactive phosphorus (TRP) concentration data collected (weekly or monthly) by the Environment Agency was used in this study, because data were widely available for most FFH sites, and TRP is largely equivalent to soluble reactive (bioavailable) phosphorus, which will be most relevant when assessing eutrophication risk. (TRP is routinely termed as ortho-phosphorus by the Environment Agency). The TRP concentrations were determined by using a molybdenum-based colorimetric methodology on an unfiltered river water sample (Murphy and Riley, 1962). Mean daily flow (Q) for each site was paired with the available TRP data. Some sites had no TRP or Q data and in some cases the flow, TRP and future flows sites were not co-located. In these instances the nearest sites on the same stretch of river were used, where appropriate.

2.4. Model development and limitations

Each TRP concentration data set was plotted as a time series to identify any sudden and obvious changes in the concentration/flow relationship and the length of data set that could be used for the LAM modelling to determine the current loading of TRP. The period from 1st January 2009 to 31st December 2014 was used to fit the LAM whenever possible. Model fitting was carried out to produce the lowest sum of square errors in each individual observation.

Model error can be explained by variable record lengths and sampling frequencies, data gaps and limits of detection issues (Supplementary Table 1). Step change reductions in TRP concentration occurred at many sites. However these were mainly before 2007 (when P stripping was widely implemented at STWs across England). Where sites show step changes in TRP concentration within the 2009 to 2014 period, the model was fitted to the latter, shorter period of stable concentrations. Some (23) sites do not have a complete set of monitoring data from 2009 to 2014. In such cases, the monitoring period was extended backward to 2007 or 2008, so that there was enough data for adequate model fitting. This was only done if there were no obvious changes in TRP concentration in the time series data, or changes in the TRP concentration/flow relationship. In some cases (36) only shorter records or earlier records existed. Some sites contain a lot of scatter or lacked high flow data (due to monthly sampling which tends to underrepresent high flow periods) (Bowes et al., 2009a, b).

The TRP data at some sites (24) were at or below the Environment Agency laboratory limits of detection (LOD; usually 0.02 mg P/L), and at some sites, this problem was compounded by the data set being comprised of a mixture of LOD values and "real" values below LOD. For sites where this was a mix of LOD values and values below LOD, all 0.02 mg/L LOD data points were removed, and the remaining data used to fit the LAM, as the data below 0.02 mg/L appeared to be reliable. At sites where the LOD values were consistently used, these 0.02 mg P/L values were used for the load apportionment modelling.

2.5. Application of LAMs to future flows

The modelled relationship between TRP concentration and flow for each site, based on the observed data over recent years, was applied to the 150 year flow projections of the Future Flows Hydrology (FFH) data sets for that site, to produce projections of TRP concentrations for 11 FFH scenarios. The future mean daily river flows provide the *Q* term in Eq. (1).

The final output from the Load Apportionment Model (LAM) applications is 11 time series of phosphorus concentration projections from 1951 to 2098, corresponding to the 11 ensemble members of the FFH dataset, The climate baseline (1961–1990) and future (2040–2069) time periods were extracted from the daily time series to provide separate annual and summer P concentration averages for these 30 year periods.

2.6. Sewage treatment scenario

To assess the impact of changes in sewage treatment, a realistic future treatment scenario was created by re-calculating the value of the

[1] http://www.ceh.ac.uk/our-science/projects/future-flows-and-groundwater-levels

Table 1
Future flows station details. BFI is the base flow index. SAAR is the average annual rainfall in the standard period (1961–1990). See, for example, www.nrfa.ceh.ac.uk/feh-catchment-descriptors.

Station details							
Station	Name	Easting	Northing	Catchment area (km^2)	BFI	Urban extent	SAAR 61–90 (mm)
21032	Glen at Kirknewton	391848	631028	198.9	0.5	0.1	876
23004	South Tyne at Haydon Bridge	385656	564671	751.1	0.34	0.2	1148
23011	Kielder Burn at Kielder	364442	594681	58.8	0.32	0	1199
24005	Browney at Burn Hall	425904	538688	178.5	0.49	3.1	743
24009	Wear at Chester le Street	428304	551226	1008.3	0.47	3.1	855
25005	Leven at Leven Bridge	444431	512072	196.3	0.42	1.4	725
25019	Leven at Easby	458466	508686	14.8	0.58	0.4	831
25020	Skerne at Preston le Skerne	429210	523780	147	0.41	4.9	654
27002	Wharfe at Flint Mill Weir	442215	447311	758.9	0.4	1.6	1161
27007	Ure at Westwick Lock	435599	467047	914.6	0.39	0.8	1118
27009	Ouse at Skelton	456845	455373	3315	0.45	1.5	900
27021	Don at Doncaster	456977	403973	1256.2	0.57	13.5	799
27034	Ure at Kilgram Bridge	419062	485989	510.2	0.32	0.4	1342
27035	Aire at Kildwick Bridge	401106	445684	282.3	0.37	2	1153
27041	Derwent at Buttercrambe	473112	458712	1586	0.7	0.8	765
27042	Dove at Kirkby Mills	470468	485533	59.2	0.61	0.8	906
27043	Wharfe at Addingham	409146	449298	427	0.34	0.4	1383
27049	Rye at Ness	469439	479196	238.7	0.68	0.3	839
27084	Eastburn Beck at Crosshills	402035	445263	43.4	0.35	1.8	1129
28008	Dove at Rocester Weir	411240	339670	399	0.62	0.7	1021
28031	Manifold at Ilam	413980	350720	148.5	0.53	0.3	1096
28033	Dove at Hollinsclough	406320	366830	8	0.48	0	1349
28046	Dove at Izaak Walton	414710	351000	83	0.79	0.4	1096
28055	Ecclesbourne at Duffield	431940	344640	50.4	0.49	2.3	853
28066	Cole at Coleshill	418170	287360	130	0.42	39.5	722
31010	Chater at Fosters Bridge	496070	303020	68.9	0.53	0.5	640
33012	Kym at Meagre Farm	515607	263134	137.5	0.26	0.7	585
33014	Lark at Temple	575805	272944	272	0.77	3.5	593
33018	Tove at Cappenham Bridge	471144	248673	138.1	0.54	1.6	661
33019	Thet at Melford Bridge	587961	283000	316	0.78	1.4	620
33026	Bedford Ouse at Offord	521661	266946	2570	0.5	4	609
33027	Rhee at Wimpole	533301	248518	119.1	0.65	1.3	558
33029	Stringside at Whitebridge	571602	300623	98.8	0.84	0.7	629
33044	Thet at Bridgham	595681	285495	277.8	0.74	1.3	620
33063	Little Ouse at Knettishall	595497	280786	101	0.65	1	595
34002	Tas at Shotesham	622583	299391	146.5	0.59	1.5	610
34006	Waveney at Needham Mill	622906	281137	370	0.47	1.4	594
34014	Wensum at Swanton Morley Total	602085	318419	397.8	0.75	1.3	684
35008	Gipping at Stowmarket	605756	257940	128.9	0.39	2.8	577
36005	Brett at Hadleigh	602441	242918	156	0.47	0.9	580
36007	Belchamp Brook at Bardfield Bridge	584785	242156	58.6	0.42	0.4	560
37001	Roding at Redbridge	541499	188348	303.3	0.39	6.9	606
37011	Chelmer at Churchend	562886	223350	72.6	0.43	1.2	591
37019	Beam at Bretons Farm	551533	185330	49.7	0.37	33.9	588
38003	Mimram at Panshanger Park	528256	213276	133.9	0.93	6.5	656
38014	Salmon Brook at Edmonton	534361	193703	20.5	0.29	29.3	666
39001	Thames at Kingston	517780	169850	9948	0.63	6.6	706
39006	Windrush at Newbridge	440179	201858	362.6	0.86	1.5	743
39034	Evenlode at Cassington Mill	444816	209933	430	0.71	1.4	691
39049	Silk Stream at Colindeep Lane	521705	189500	29	0.33	40.1	685
39057	Crane at Cranford Park	510312	177840	61.7	0.35	48.9	639
39076	Windrush at Worsham	430140	210658	296	0.82	0.7	763
39081	Ock at Abingdon	448148	196667	234	0.64	1.8	639
39090	Cole at Inglesham	420820	196950	140	0.52	6.5	682
39096	Wealdstone Brook at Wembley	519359	186216	21.8	0.24	50.8	664
39105	Thame at Wheatley	461190	205030	533.8	0.55	3.6	644
39131	Brent at Costons Lane Greenford	514914	182238	146.2	0.29	52.8	664
40003	Medway at Teston	570865	153033	1256.1	0.4	3.4	744
40011	Great Stour at Horton	611549	155356	345	0.7	3.2	747
40017	Dudwell at Burwash	567860	124040	27.5	0.43	1	888
40023	East Stour at South Willesborough	601513	140704	58.8	0.45	1.5	766
41011	Rother at Iping Mill	485220	122904	154	0.67	2.9	920
41022	Lod at Halfway Bridge	493124	122259	52	0.35	0.9	858
41026	Cockhaise Brook at Holywell	537653	126163	36.1	0.52	1	851
42012	Anton at Fullerton	437890	139230	185	0.96	3.6	773
43003	Avon at East Mills	415868	114355	1477.8	0.91	1.6	807
43005	Avon at Amesbury	415109	141387	323.7	0.91	1.3	745
43006	Nadder at Wilton	409725	130794	220.6	0.81	0.9	875
43007	Stour at Throop	411233	96046	1073	0.64	2	861
43021	Avon at Knapp Mill	415607	94304	1706	0.86	1.7	810
44002	Piddle at Baggs Mill	391322	87609	183.1	0.89	0.5	943
45001	Exe at Thorverton	293602	101602	600.9	0.5	0.6	1248

(continued on next page)

Table 1 (*continued*)

Station details							
Station	Name	Easting	Northing	Catchment area (km^2)	BFI	Urban extent	SAAR 61–90 (mm)
45004	Axe at Whitford	326208	95324	288.5	0.47	1.1	994
45005	Otter at Dotton	308665	88435	202.5	0.53	2.4	976
47001	Tamar at Gunnislake	242627	72524	916.9	0.46	0.5	1216
47008	Thrushel at Tinhay	239783	85516	112.7	0.43	0.1	1143
47014	Walkham at Horrabridge	251312	69888	44.6	0.58	0.8	1666
48003	Fal at Tregony	192107	44747	87	0.67	1.7	1210
49001	Camel at Denby	201748	68159	208.8	0.62	1.2	1336
50002	Torridge at Torrington	249955	118564	663	0.38	0.4	1186
50006	Mole at Woodleigh	266011	121039	327.5	0.47	0.4	1307
50007	Taw at Taw Bridge	267293	106820	71.4	0.46	0.6	1236
51001	Doniford Stream at Swill Bridge	308848	142865	75.8	0.66	1	908
52010	Brue at Lovington	358994	131756	135.2	0.47	1.1	867
53005	Midford Brook at Midford	376315	161132	147.4	0.62	4.4	965
53006	Frome (Bristol) at Frenchay	363753	177202	148.9	0.38	11.4	792
53017	Boyd at Bitton	368142	169866	47.9	0.44	1.6	808
53018	Avon at Bathford	378533	167023	1552	0.57	3	817
54001	Severn at Bewdley	378230	276160	4325	0.53	2	913
54008	Teme at Tenbury	359770	268520	1134.4	0.55	0.6	841
54036	Isbourne at Hinton on the Green	402368	240680	90.7	0.56	1.2	704
54038	Tanat at Llanyblodwel	325230	322440	229	0.48	0.1	1290
54057	Severn at Haw Bridge	384491	227878	9895	0.56	3.5	792
55002	Wye at Belmont	348500	238799	1895.9	0.46	0.3	1231
55003	Lugg at Lugwardine	354871	240585	885.8	0.64	0.5	812
68001	Weaver at Ashbrook	367171	363507	622	0.54	2.7	731
68005	Weaver at Audlem	365254	343040	207	0.54	0.7	719
71001	Ribble at Samlesbury	358922	430412	1145	0.34	3.7	1353
71006	Ribble at Henthorn	372190	439170	456	0.31	1.5	1348
71009	Ribble at New Jumbles Rock	370249	437592	1053	0.33	3.9	1370
72004	Lune at Caton	352935	465318	983	0.32	0.4	1523
72014	Conder at Galgate	348160	455371	28.5	0.35	0.6	1181
72015	Lune at Lunes Bridge	361210	502901	141.5	0.32	0.2	1632
73005	Kent at Sedgwick	350877	487421	209	0.41	1.8	1732
73009	Sprint at Sprint Mill	351477	496106	34.6	0.32	0	2018
73011	Mint at Mint Bridge	352411	494470	65.8	0.37	0.1	1604
73013	Rothay at Miller Bridge House	337125	504195	64	0.31	0.4	2387
73014	Brathay at Jeffy Knotts	335965	503406	57.4	0.28	0	2754
74001	Duddon at Duddon Hall	319526	489585	85.7	0.29	0	2265
74005	Ehen at Braystones	300909	506051	125.5	0.43	1.1	1758
74007	Esk at Cropple How	313100	497770	70.2	0.3	0	2305
75017	Ellen at Bullgill	309600	538400	96	0.49	0.6	1110
76005	Eden at Temple Sowerby	360452	528316	616.4	0.37	0.4	1146
76007	Eden at Sheepmount	339000	557103	2286.5	0.49	0.8	1183
76008	Irthing at Greenholme	348619	558073	334.6	0.32	0.3	1073

A parameter in Eq. (1) for each site, to represent the loading from all sewage treatment works (STWs) upstream within the associated waterbody discharging at the current volumetric rate but with a final effluent concentration reduced to 0.5 mg-P/L. This figure was chosen because current technology can deliver this level of treatment at larger works (Carey and Migliaccio, 2009). The calculation was carried out as follows:

- The total population equivalent of all STWs in the river waterbody was calculated. This is a metric calculated on the basis of all consented discharges served by the works.
- The total dry weather flow (DWF) (i.e. the flow of effluent through a STW during a sustained period of dry weather, under minimum influence of rainwater/infiltration) associated with this population equivalent was calculated on the basis of an assumed discharge of 180 L/person/day and compared with independent estimates of this upstream discharge.
- For sites where the concentration is currently above 0.5 mg-P/L, the STW load was calculated by multiplying the calculated discharge volume by the assumed concentration (0.5 mg-P/L).

This allowed the percentage reduction in STW P loading to the river under this improvement scenario to be estimated. The resulting daily P concentrations for each of the eleven FFH scenarios were calculated. In cases where the re-calculated *A* parameter (see Eq. (1)) was larger in magnitude than the baseline value, the original baseline was retained (i.e. the future treatment scenario was assumed identical to the present day, in terms of point source discharges); this was the case for 36% of sites.

2.7. Applying WFD P standards

The estimated P concentrations from the LAM for the 11 climate change scenarios were assessed against P standards introduced under the European Water Framework Directive (UKTAG[2]; DEFRA, 2014). The UK standards, based on alkalinity and altitude data, reflect natural variations in nutrient concentrations along and between rivers and have been calculated by the Environment Agency at WFD water quality monitoring locations. We used the monitoring location closest to our FFH sites which are spread across a range of alkalinities and altitude.

3. Results

3.1. Modelling present and future phosphorus concentrations

3.1.1. Load Apportionment Models

The LAM was able to produce realistic model fits to most of the data sets (see Fig. 2 for examples and Table 2 for details) and plausible estimates of the relative TRP loads from STW and diffuse, rain-related

[2] UKTAG website: http://www.wfduk.org/

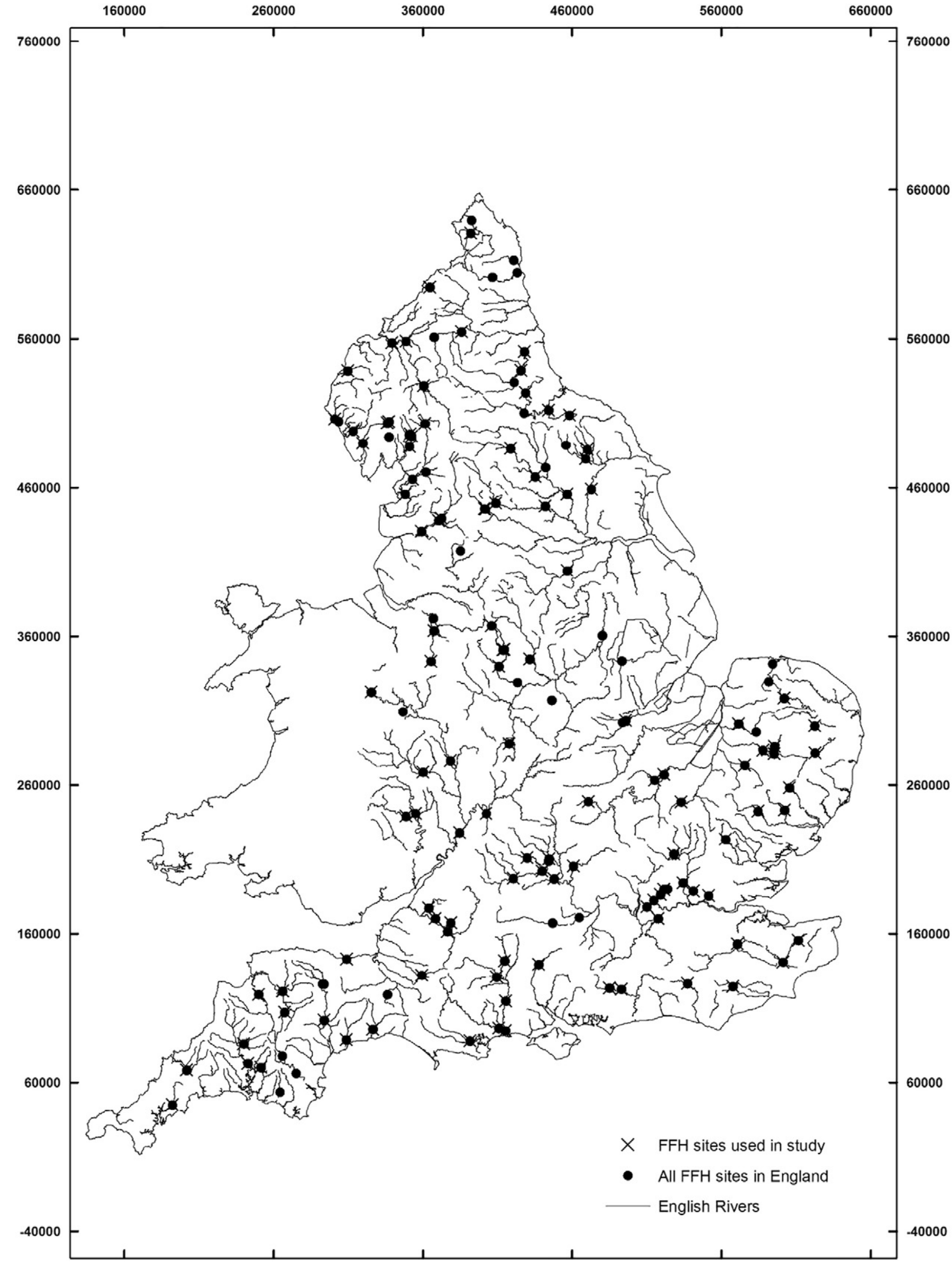

Fig. 1. Location of Future Flows Hydrology sites used in this study (X) and those not used.

sources. Table 2 lists the model parameters for each site. Map investigations for a subset of study sites confirmed that the model was correctly identifying sites with large likely STW inputs (Rivers Don and Medway; Fig. 2), mixed contributions from STW and diffuse sources (Rivers Nadder and Avon) and without any significant STW inputs (the Rivers Eden and Glen; Fig. 2).

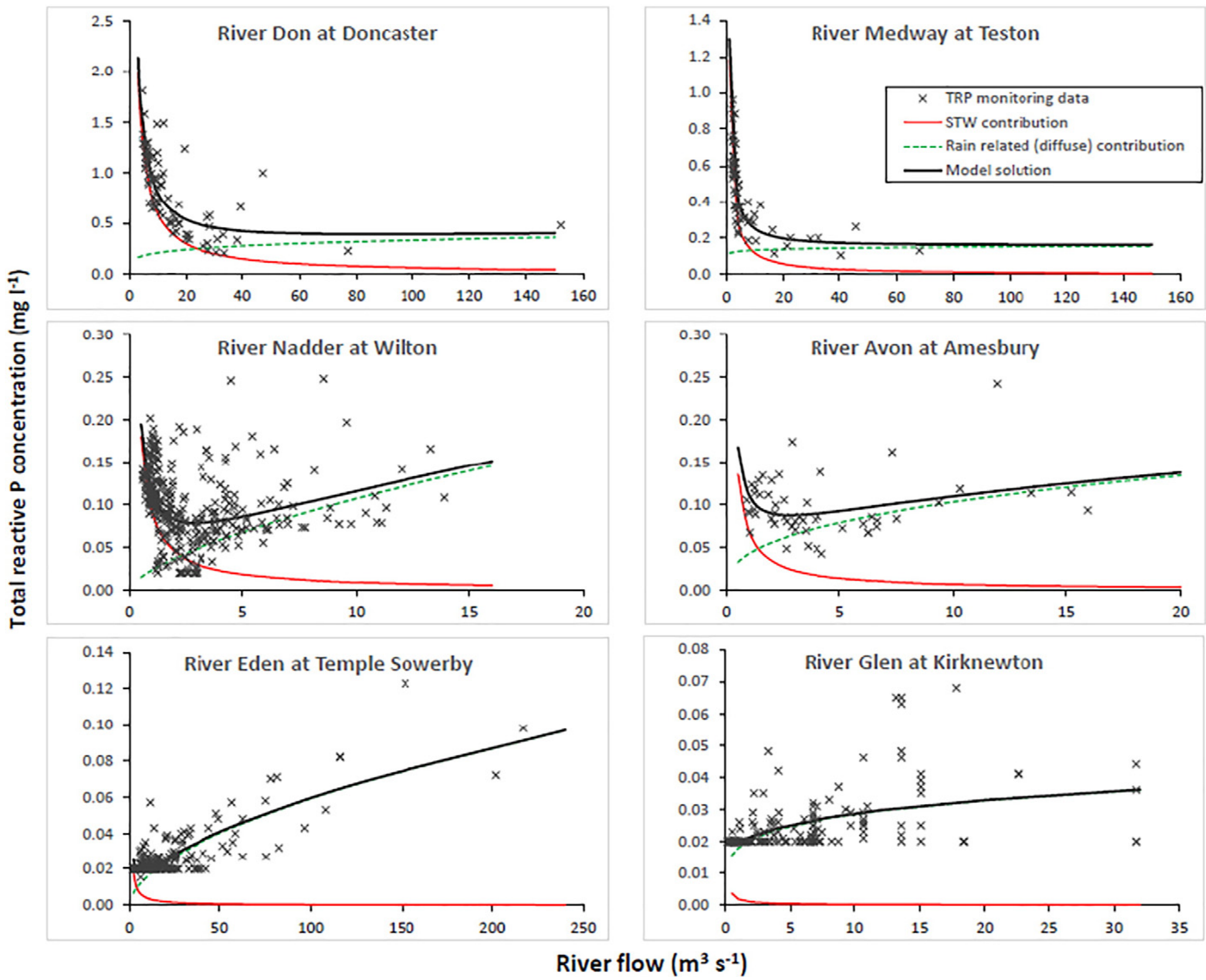

Fig. 2. Total reactive P concentration/flow relationships and Load Apportionment Model fits for a selection of sites covering a range of sewage treatment works inputs. The River Eden and River Glen examples show impact of 0.02 mg P/L "limit of detection" observations.

In this study 88% of sites are dominated by diffuse, rain-related source contributions (>50% contribution; Table 2; Fig. 3), in terms of annual load. About 39% of sites have diffuse source contributions above 90% which indicates either that there is limited point source input at these locations or that they already have effective treatment of these point sources.

3.2. Changes in phosphorus concentrations

3.2.1. Baseline phosphorus estimates

Baseline (1961–1990) estimates of absolute annual average TRP concentrations vary considerably between sites, ranging from about 0.001 to about 3.350 mg/L (Fig. 4 shows the maximum, median and minimum values for each site across the 11 ensemble members). Relatively high TRP concentrations are found around London and Bristol, in the North East south of the River Tyne, and to a lesser extent scattered across the Midlands and East Anglia. There is little variation between flow scenarios with differences between maximum, median and minimum estimates relatively small except in some locations. At 97% of sites the range between maximum and minimum estimates is below 0.1 mg/L. Only at the River Chelmer at Churchend is the range significantly larger (0.75 mg/L); at three other locations (River Medway, Midford Brook and River Weaver) estimates of range are marginally above 0.1 mg TRP/L. Using maximum, median and minimum captures the uncertainty across the flow scenarios but it should be noted that the minimum, median and maximum ensemble member values can be obtained from different members at different sites. This indicates limited impact of flow uncertainty on P estimates. There is a slight increase in summer TRP values compared to annual TRP values (Fig. 4). This implies that these sites are dominated by constant STW inputs through the summer low-flow period.

We have classified the results according to the maximum site-specific TRP WFD status boundaries across the data set calculated for each site. This enables a simple comparison across the varied WFD standard values. Approximately 35% of annual estimates are below 0.055 mg TRP/L (although this is slightly higher for annual minimum). Between 25 and 30% of estimates are above 0.227 mg/L when averaged across the year; this increases to between about 30–35% in the summer.

3.2.2. Future phosphorus estimates

Climate and flow-related changes in projected TRP concentrations mostly show increases from the Baseline to 2050s period – both as absolute mg/L values and also as percentage change relative to the baseline (Fig. 5), with a few site exceptions where TRP concentrations are

Table 2
Model parameters, fit and diffuse source contribution. Ascen is the revised A parameter re-calculated for the sewage treatment scenario.

Station		Model parameters and fit						Diffuse contribution (%)
Station	Name	Anow	Cnow	Dnow	SSQ	Qcross	Ascen	
21032	Glen at Kirknewton	0.002	0.018	1.202	0.011	0.148	0.002	98.8
23004	South Tyne at Haydon Bridge	0.003	0.023	1.064	0.008	0.163	0.003	99.2
23011	Kielder Burn at Kielder	0.000	0.021	1.040	0.001	0.000	0.000	100.0
24005	Browney at Burn Hall	0.092	0.137	1.000	0.563	0.675	0.056	73.8
24009	Wear at Chester le Street	1.422	0.169	1.000	2.129	8.436	0.059	69.9
25005	Leven at Leven Bridge	0.059	0.134	1.015	0.159	0.445	0.020	79.1
25019	Leven at Easby	0.000	0.041	1.000	0.112	0.004	0.000	98.2
25020	Skerne at Preston le Skerne	0.077	0.075	1.265	0.069	1.021	0.024	63.9
27002	Wharfe at Flint Mill Weir	0.496	0.013	1.000	0.020	37.998	0.103	32.8
27007	Ure at Westwick Lock	0.000	0.024	1.008	0.003	0.000	0.000	100.0
27009	Ouse at Skelton	0.586	0.060	1.000	0.060	9.772	0.255	85.6
27021	Don at Doncaster	5.936	0.129	1.205	3.856	23.970	1.701	40.9
27034	Ure at Kilgram Bridge	0.002	0.018	1.041	0.000	0.117	0.002	99.4
27035	Aire at Kildwick Bridge	0.462	0.008	1.546	0.113	13.535	0.030	50.0
27041	Derwent at Buttercrambe	0.291	0.008	1.458	0.022	11.483	0.086	64.7
27042	Dove at Kirkby Mills	0.000	0.020	1.003	0.000	0.000	0.000	100.0
27043	Wharfe at Addingham	0.000	0.020	1.000	0.000	0.000	0.000	100.0
27049	Rye at Ness	0.002	0.020	1.000	0.001	0.114	0.002	96.9
27084	Eastburn Beck at Crosshills	0.003	0.046	1.276	0.023	0.123	0.000	95.2
28008	Dove at Rocester Weir	0.043	0.019	1.247	0.001	1.917	0.006	77.3
28031	Manifold at Ilam	0.000	0.034	1.058	0.029	0.000	0.000	100.0
28033	Dove at Hollinsclough	0.004	0.021	1.000	0.035	0.216	0.001	65.1
28046	Dove at Izaak Walton	0.000	0.067	1.435	0.783	0.000	0.000	100.0
28055	Ecclesbourne at Duffield	0.029	0.080	1.124	0.061	0.408	0.008	64.1
28066	Cole at Coleshill	0.049	0.077	1.222	0.264	0.693	0.049	77.3
31010	Chater at Fosters Bridge	0.044	0.105	1.639	0.231	0.583	0.002	21.5
33012	Kym at Meagre Farm	0.020	0.170	1.000	2.042	0.120	0.004	84.9
33014	Lark at Temple	0.052	0.083	1.000	0.103	0.624	0.052	64.4
33018	Tove at Cappenham Bridge	0.083	0.052	1.000	0.461	1.597	0.026	36.1
33019	Thet at Melford Bridge	0.008	0.072	1.000	0.074	0.115	0.008	93.6
33026	Bedford Ouse at Offord	0.561	0.128	1.000	0.261	4.372	0.561	80.3
33027	Rhee at Wimpole	0.022	0.150	1.000	0.874	0.148	0.022	84.9
33029	Stringside at Whitebridge	0.003	0.035	1.000	0.012	0.072	0.001	88.4
33044	Thet at Bridgham	0.039	0.071	1.000	0.080	0.550	0.023	72.9
33063	Little Ouse at Knettishall	0.005	0.068	1.000	0.035	0.080	0.004	86.5
34002	Tas at Shotesham	0.037	0.098	1.102	0.017	0.409	0.011	79.7
34006	Waveney at Needham Mill	0.000	0.106	1.118	0.086	0.000	0.000	100.0
34014	Wensum at Swanton Morley Total	0.030	0.068	1.058	0.151	0.463	0.030	83.9
35008	Gipping at Stowmarket	0.013	0.154	1.000	0.227	0.082	0.005	85.4
36005	Brett at Hadleigh	0.026	0.066	1.009	0.074	0.403	0.011	66.7
36007	Belchamp Brook at Bardfield Bridge	0.000	0.060	1.004	0.016	0.000	0.000	100.0
37001	Roding at Redbridge	0.120	0.202	1.000	0.994	0.592	0.031	73.5
37011	Chelmer at Churchend	0.043	0.157	1.000	1.103	0.270	0.014	54.8
37019	Beam at Bretons Farm	0.032	0.107	1.492	3.441	0.448	0.032	44.8
38003	Mimram at Panshanger Park	0.008	0.051	1.485	0.002	0.279	0.002	77.4
38014	Salmon Brook at Edmonton	0.003	0.218	1.000	0.733	0.014	0.003	92.7
39001	Thames at Kingston	2.392	0.320	1.000	0.720	7.469	2.392	86.2
39006	Windrush at Newbridge	0.118	0.001	3.095	0.085	4.465	0.039	15.4
39034	Evenlode at Cassington Mill	0.179	0.028	1.331	0.159	4.029	0.040	38.5
39049	Silk Stream at Colindeep Lane	0.037	0.172	1.141	1.292	0.260	0.037	48.4
39057	Crane at Cranford Park	0.013	0.181	1.000	0.416	0.070	0.013	88.6
39076	Windrush at Worsham	0.079	0.000	4.793	0.016	5.473	0.010	43.2
39081	Ock at Abingdon	0.092	0.117	1.000	0.175	0.784	0.046	62.5
39090	Cole at Inglesham	0.046	0.097	1.072	0.278	0.497	0.006	60.1
39096	Wealdstone Brook at Wembley	0.013	0.263	1.298	1.157	0.097	0.013	59.4
39105	Thame at Wheatley	0.799	2.637	1.000	13.524	0.303	0.179	94.6
39131	Brent at Costons Lane Greenford	0.062	0.273	1.000	0.248	0.226	0.062	81.7
40003	Medway at Teston	1.180	0.118	1.057	0.824	8.857	0.465	49.2
40011	Great Stour at Horton	0.224	0.038	1.438	0.304	3.442	0.152	54.8
40017	Dudwell at Burwash	0.001	0.075	2.131	0.071	0.153	0.001	90.8
40023	East Stour at South Willesborough	0.006	0.131	1.200	0.449	0.081	0.005	96.2
41011	Rother at Iping Mill	0.140	0.153	1.000	0.131	0.915	0.060	72.6
41022	Lod at Halfway Bridge	0.004	0.065	1.000	0.023	0.060	0.004	91.2
41026	Cockhaise Brook at Holywell	0.002	0.032	1.000	0.024	0.056	0.001	85.7
42012	Anton at Fullerton	0.005	0.027	1.000	0.060	0.179	0.005	91.3
43003	Avon at East Mills	0.472	0.005	1.732	0.293	13.105	0.125	61.5
43005	Avon at Amesbury	0.067	0.043	1.382	0.055	1.396	0.019	84.6
43006	Nadder at Wilton	0.090	0.023	1.661	0.476	2.256	0.006	65.4
43007	Stour at Throop	1.608	0.028	1.478	0.089	15.343	0.403	62.8
43021	Avon at Knapp Mill	0.443	0.008	1.633	0.055	11.982	0.150	67.7
44002	Piddle at Baggs Mill	0.000	0.039	1.082	0.036	0.000	0.000	100.0
45001	Exe at Thorverton	0.185	0.008	1.276	0.022	11.654	0.037	64.0
45004	Axe at Whitford	0.041	0.063	1.185	0.052	0.698	0.015	91.7

(continued on next page)

Table 2 (*continued*)

Station		Model parameters and fit						Diffuse contribution (%)
Station	Name	Anow	Cnow	Dnow	SSQ	Qcross	Ascen	
45005	Otter at Dotton	0.122	0.088	1.300	0.107	1.286	0.008	78.2
47001	Tamar at Gunnislake	0.103	0.010	1.302	0.002	5.877	0.003	91.0
47008	Thrushel at Tinhay	0.003	0.017	1.252	0.001	0.253	0.001	94.6
47014	Walkham at Horrabridge	0.011	0.004	1.888	0.002	1.675	0.003	84.3
48003	Fal at Tregony	0.038	0.021	1.138	0.030	1.658	0.002	61.2
49001	Camel at Denby	0.135	0.038	1.000	0.183	3.562	0.006	65.2
50002	Torridge at Torrington	0.000	0.039	1.000	0.010	0.000	0.000	100.0
50006	Mole at Woodleigh	0.014	0.010	1.302	0.002	1.259	0.007	95.9
50007	Taw at Taw Bridge	0.072	0.027	1.000	1.416	2.674	0.003	42.0
51001	Doniford Stream at Swill Bridge	0.025	0.031	2.162	0.081	0.895	0.003	88.8
52010	Brue at Lovington	0.119	0.101	1.243	0.169	1.139	0.013	78.0
53005	Midford Brook at Midford	0.388	0.033	1.453	1.067	5.471	0.045	33.1
53006	Frome (Bristol) at Frenchay	0.018	0.084	1.220	0.020	0.275	0.018	92.1
53017	Boyd at Bitton	0.027	0.182	1.000	1.206	0.147	0.003	76.4
53018	Avon at Bathford	0.837	0.053	1.214	0.250	9.698	0.814	71.1
54001	Severn at Bewdley	1.469	0.074	1.000	0.063	19.897	0.446	71.6
54008	Teme at Tenbury	0.039	0.024	1.154	0.016	1.506	0.029	91.9
54036	Isbourne at Hinton on the Green	0.064	0.082	1.220	0.100	0.814	0.007	55.9
54038	Tanat at Llanyblodwel	0.029	0.021	1.290	0.079	1.265	0.001	93.2
54057	Severn at Haw Bridge	4.807	0.145	1.000	0.437	33.258	2.641	76.6
55002	Wye at Belmont	0.005	0.033	1.062	0.054	0.160	0.005	99.7
55003	Lugg at Lugwardine	0.109	0.012	1.519	0.050	4.257	0.023	84.8
68001	Weaver at Ashbrook	0.862	0.355	1.000	2.229	2.429	0.141	69.7
68005	Weaver at Audlem	0.044	0.213	1.000	0.328	0.208	0.003	90.7
71001	Ribble at Samlesbury	1.368	0.095	1.052	4.652	12.634	0.448	74.9
71006	Ribble at Henthorn	0.000	0.053	1.083	0.162	0.000	0.000	100.0
71009	Ribble at New Jumbles Rock	0.000	0.028	1.183	0.123	0.000	0.000	100.0
72004	Lune at Caton	0.000	0.010	1.197	0.004	0.000	0.000	100.0
72014	Conder at Galgate	0.001	0.085	1.145	0.054	0.015	0.001	98.7
72015	Lune at Lunes Bridge	0.010	0.009	1.147	0.001	1.039	0.001	91.1
73005	Kent at Sedgwick	0.084	0.006	1.335	0.010	7.169	0.064	70.8
73009	Sprint at Sprint Mill	0.000	0.003	1.380	0.000	0.300	0.000	96.3
73011	Mint at Mint Bridge	0.001	0.010	1.335	0.002	0.238	0.000	97.7
73013	Rothay at Miller Bridge House	0.003	0.002	1.360	0.001	1.400	0.003	83.7
73014	Brathay at Jeffy Knotts	0.002	0.004	1.000	0.000	0.567	0.002	87.3
74001	Duddon at Duddon Hall	0.000	0.001	1.371	0.000	0.448	0.000	97.1
74005	Ehen at Braystones	0.000	0.009	1.296	0.007	0.000	0.000	100.0
74007	Esk at Cropple How	0.000	0.001	1.095	0.000	0.014	0.000	99.8
75017	Ellen at Bullgill	0.191	0.009	1.851	0.220	5.220	0.009	33.1
76005	Eden at Temple Sowerby	0.038	0.004	1.563	0.014	3.961	0.014	94.3
76007	Eden at Sheepmount	0.000	0.012	1.268	0.147	0.000	0.000	100.0
76008	Irthing at Greenholme	0.010	0.012	1.297	0.002	0.918	0.006	94.1

reduced in some projections. Compared to the baseline period, the projections of annual average TRP concentrations typically show small increases in the 2050s. Like the baseline, annual average TRP projections for the 2050s are very variable between sites but relatively uniform between climate scenarios. For annual TRP concentrations, median change ranges from −7% (River Chelmer) to 32% (Midford Brook), with 82% of sites increasing. The maximum change is +68% (River Windrush) whilst the minimum change is −29% (River Chelmer). The River Windrush value is the extreme value across all sites and all ensemble members – there are no other changes above 50% - and is the result of

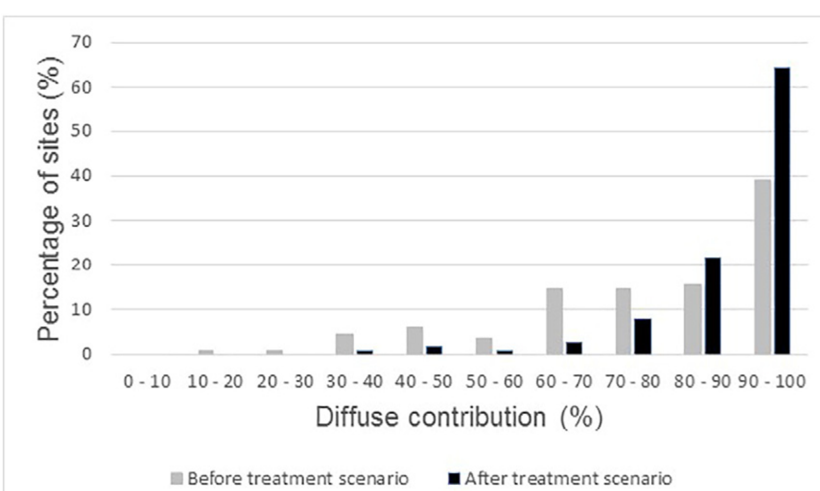

Fig. 3. Frequency distribution of percentage diffuse contributions across sites under existing sewage treatment levels and using a scenario stripping P by up to 0.5 mg/L.

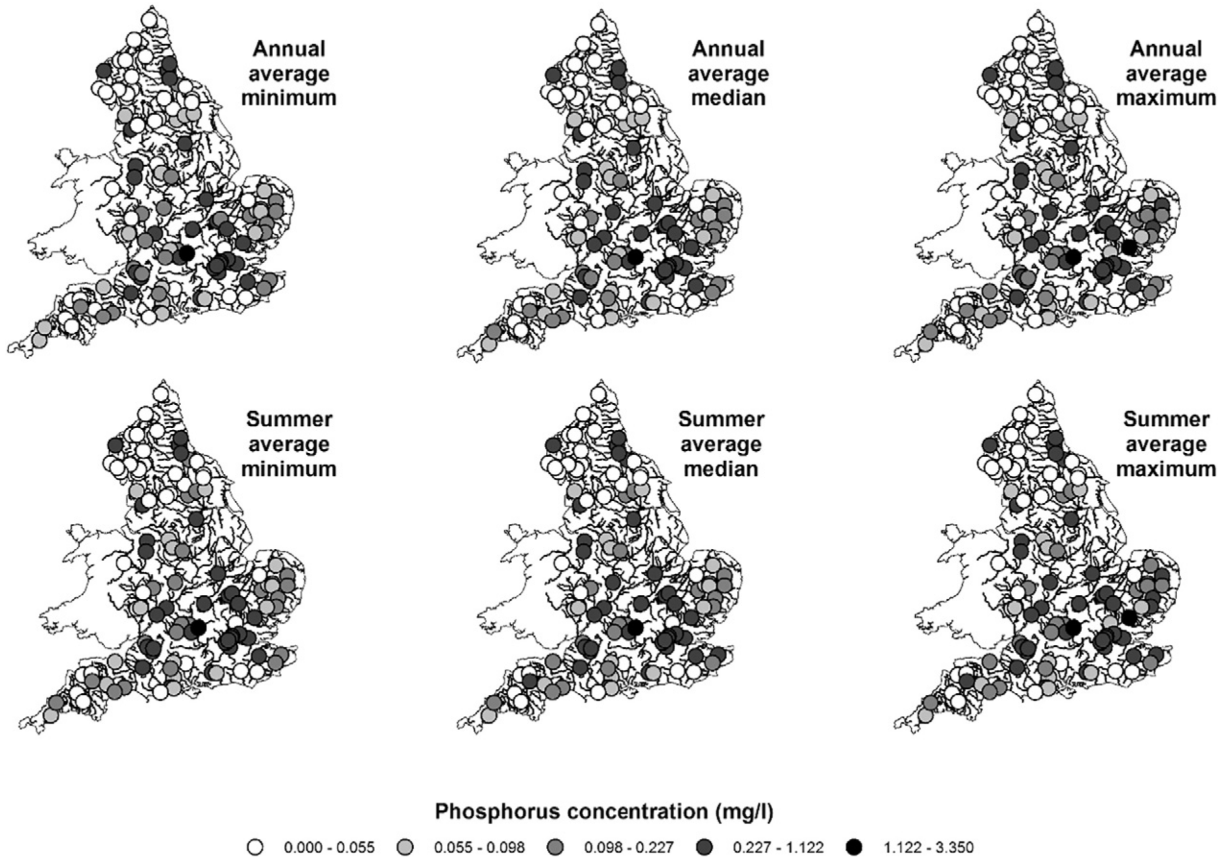

Fig. 4. (a) Maximum, median, and minimum maps of baseline absolute annual average phosphorus concentration. (b) Maximum, median, and minimum maps of baseline absolute summer average phosphorus concentration.

some very high flow values in one ensemble member in the 2050s period. For maximum TRP concentrations, 97% of site estimates increase, whilst for the minimum only 35% increase. This indicates some uncertainty in the direction of P concentration increases at some locations, although these reductions tend to be much smaller than the estimated increases. There is limited consistency in the spatial distribution of percentage change in P between the baseline and the 2050s although there is some indication that there are more reductions in TRP concentration projected in East Anglia and the North West compared to other areas, but most sites show an increase. These results are consistent with an earlier study for the Anglian region (Atkins, 2014) which suggested an increase in phosphorus concentration for the majority of rivers. There is a consistent pattern of greater increases for the median and maximum ensemble results in the South West. This occurs because the South West shows the most consistent decreases in river flows across ensembles than other parts of the country, especially in summer and autumn (see the maps in Prudhomme et al., 2012b).

Summer averages in the 2050s are typically higher (Fig. 5) compared to annual average TRP concentrations. Absolute changes in flow-related projections from the baseline to the 2050s period are more marked in the summer months than for the annual averages. For summer TRP concentrations, median change ranges from −8% (River Chelmer) to 36% (River Walkham), with 79% of sites increasing and a slight tendency towards greater increase in median change across the ensemble. There is also an increase in the overall range across the ensemble with a slight decrease in the minimum to −35% (Chelmer) and a large increase in the maximum up to 126% (River Windrush). When there is a decrease in summer P, this tends to be greater than the decrease in annual P. This indicates that there is the potential for significant increases in summer TRP concentrations into the future but that there is increased uncertainty around this.

3.3. Changes in WFD phosphorus status

WFD Phosphorus Status classification projections based on site specific thresholds (Table 3) change little to the 2050s (Fig. 6), although there are reductions in status for 3 sites in the south east, all of which drop from moderate (2) to poor (1). Importantly, the maps show that current flow and TRP relationships result in frequent failure of the WFD status throughout England and climate change exacerbates this pattern. It is also notable that in the North West climate change does not appear to reduce WFD status for phosphorus. In general, however, these projections suggest that further management intervention is necessary to improve WFD status for phosphorus.

3.4. Changes under STW treatment scenario

3.4.1. Future scenario

The P concentrations and WFD status estimates presented in earlier sections were produced assuming that the phosphorus concentration – flow relationships established through calibration against recent monitored data remain unchanged; i.e. the TRP inputs from diffuse catchment and point STW sources stay as they are now. Here we model changes based on the assumption that all STWs could be equipped to reduce final effluent TRP concentrations to a maximum of 0.5 mg/L. For monitored catchments where the average inputs from STWs are currently higher than this 'theoretically achievable P stripping concentration', these projections demonstrate the potential reductions associated with such a treatment intervention. A revised flow – phosphorus equation reflecting the lower STW inputs is used in association with the same 11 climate and flow scenarios. The impact of this change is an increase in the number of sites that are diffuse source dominated

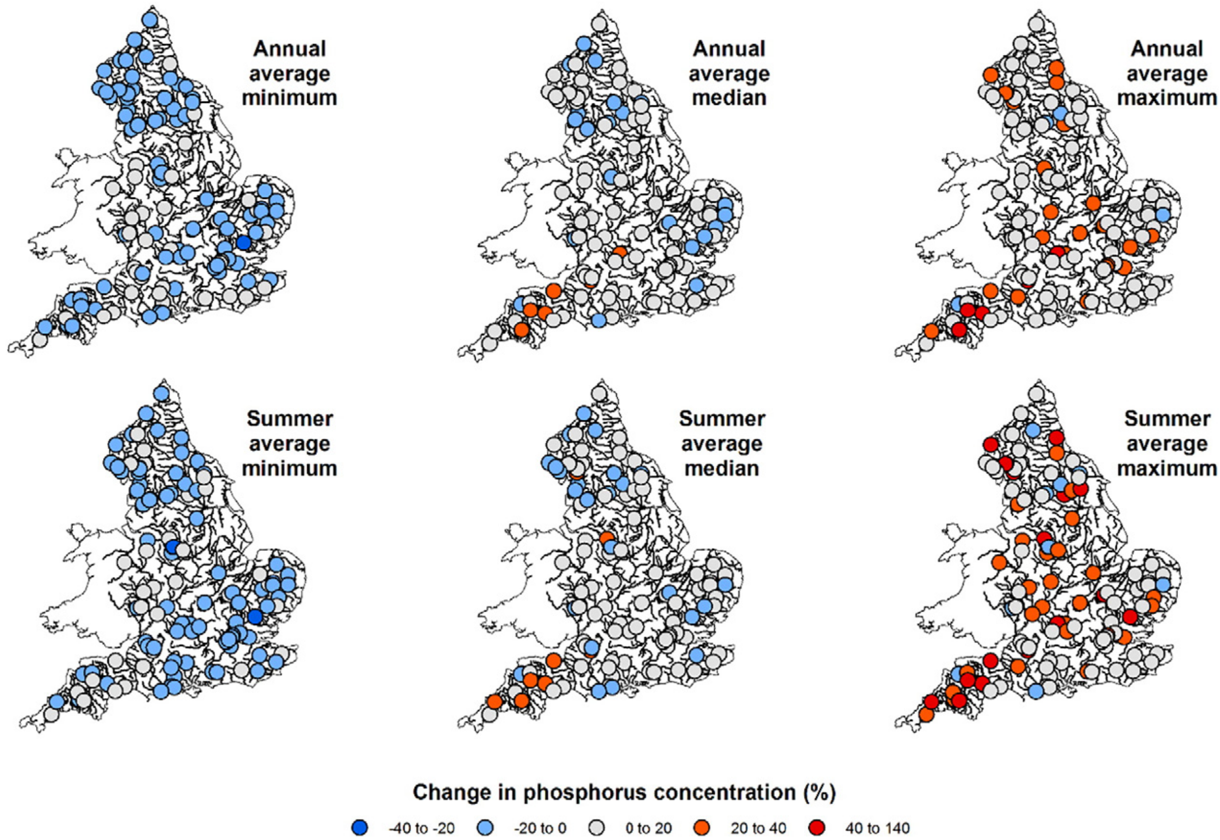

Fig. 5. (a) Maximum, median, and minimum maps of percentage change in phosphorus concentration from baseline to 2050s for annual average. (b) Maximum, median, and minimum maps of percentage change in phosphorus concentration from baseline to 2050s for summer average.

from 88% to 97%. At 64% of sites diffuse source contributions are over 90%, indicating that such a treatment scenario could address many existing point source contributions.

Under the STW treatment scenario, median TRP projections for the 2050s are much lower at many sites (Fig. 7a), being up to 0.6 mg/L lower (at Midford Brook). Approximately 50% of sites see reductions between 0 and 20% with the assumed additional phosphorus stripping at STWs mapped in Fig. 7b. Associated median WFD Status projections suggest considerable improvements would be realised by such intervention (Fig. 7c and d) at many sites (with ca. 40 sites improving), although further work would still be needed to achieve Good Status everywhere – probably associated with action to reduce diffuse catchment sources of nutrient inputs. Only one site (Ellen at Bullgill) improves by 3 WFD classifications. A further 7 (~8%) improve by 2 classifications, 34 (~30%) improve by only 1 classification, the remainder (~64%) do not improve. Although P stripping to 0.5 mg/L is effective at reducing point source contributions, it still results in limited improvement in WFD P status either now or in the 2050s.

3.4.2. Additional reductions to achieve good P status

A further scenario was assessed to determine how much reduction in P concentrations would be required to achieve current P standards in the future. The additional reductions in TRP required to reach Good Status thresholds in the 2050s are relatively small (Fig. 8a) when compared to the original projections, affecting over half of the studied sites but there are some substantial additional reductions necessary, particularly around London and parts of the Midlands. These are reduced further if the STW P stripping Scenario were to be realised as shown in Fig. 8b.

4. Discussion

Our results indicate that we should expect significant changes in future phosphorus concentrations, particularly in summer as a result of climate driven changes in river flow. Even with further investment in wastewater treatment, some sites will not meet WFD status objectives for P standards. The projected increases in P concentrations by the 2050s are variable spatially and across climate change ensemble members. A number of factors such as catchment characteristics, seasonality or uncertainty or error in the flow – P relationship could explain this variation and are discussed here as they could affect any attempts to mitigate future changes.

Catchment characteristics may influence the changes in P concentrations by altering flow patterns or the delivery of P to rivers. We plotted minimum, median, maximum annual and summer percentage change against the catchment characteristics for each site (base flow index (BFI), rainfall, urban extent, and diffuse percentage P). There are no clear associations except that as diffuse contributions increase, the maximum percentage change in Annual P decreases ($R^2 = 0.474$). This suggests a weak indication of change dependent on the nature of the catchment (whether it is diffuse or point source dominated). This weak signal may be because 88% of the sites are diffuse source dominated in terms of annual TRP load, which may reflect the nature of the stations used in Future Flows, rather than being typical of rivers in England. It also means that improvements in phosphorus concentrations requires management other than point source reductions.

The results indicate in the coming decades, the greatest increases in P concentrations will be in the summer as a result of lower river flows. This fits with other studies that suggest increased risk of eutrophication is expected following drought (e.g. Zwolsman and Van Bokhoven, 2007;

Table 3
Site specific WFD phosphorus status boundaries (mg/L).

Station		WFD P status boundary value (mg/L)			
Station	Name	Poor	Moderate	Good	High
21032	Glen at Kirknewton	0.824	0.108	0.037	0.018
23004	South Tyne at Haydon Bridge	0.851	0.116	0.041	0.020
23011	Kielder Burn at Kielder	0.779	0.094	0.031	0.014
24005	Browney at Burn Hall	1.009	0.175	0.070	0.037
24009	Wear at Chester le Street	1.112	0.222	0.095	0.053
25005	Leven at Leven Bridge	1.066	0.200	0.083	0.045
25019	Leven at Easby	0.973	0.161	0.062	0.032
25020	Skerne at Preston le Skerne	1.047	0.192	0.078	0.042
27002	Wharfe at Flint Mill Weir	1.113	0.222	0.095	0.053
27007	Ure at Westwick Lock	1.112	0.222	0.095	0.053
27009	Ouse at Skelton	1.049	0.193	0.079	0.043
27021	Don at Doncaster	1.039	0.188	0.077	0.041
27034	Ure at Kilgram Bridge	0.942	0.149	0.056	0.029
27035	Aire at Kildwick Bridge	0.919	0.140	0.052	0.026
27041	Derwent at Buttercrambe	1.096	0.214	0.091	0.050
27042	Dove at Kirkby Mills	0.902	0.134	0.049	0.025
27043	Wharfe at Addingham	0.885	0.128	0.046	0.023
27049	Rye at Ness	1.043	0.190	0.078	0.042
27084	Eastburn Beck at Crosshills	0.919	0.140	0.052	0.026
28008	Dove at Rocester Weir	1.004	0.173	0.069	0.036
28031	Manifold at Ilam	0.936	0.146	0.055	0.028
28033	Dove at Hollinsclough	0.833	0.111	0.038	0.018
28046	Dove at Izaak Walton	0.964	0.157	0.061	0.031
28055	Ecclesbourne at Duffield	0.965	0.158	0.061	0.031
28066	Cole at Coleshill	0.992	0.168	0.066	0.035
31010	Chater at Fosters Bridge	1.057	0.196	0.081	0.044
33012	Kym at Meagre Farm	1.115	0.223	0.096	0.053
33014	Lark at Temple	1.121	0.226	0.097	0.054
33018	Tove at Cappenham Bridge	1.038	0.188	0.076	0.041
33019	Thet at Melford Bridge	1.122	0.227	0.098	0.055
33026	Bedford Ouse at Offord	1.103	0.218	0.093	0.051
33027	Rhee at Wimpole	1.108	0.220	0.094	0.052
33029	Stringside at Whitebridge	1.117	0.224	0.096	0.054
33044	Thet at Bridgham	1.114	0.223	0.095	0.053
33063	Little Ouse at Knettishall	1.113	0.222	0.095	0.053
34002	Tas at Shotesham	1.115	0.223	0.096	0.053
34006	Waveney at Needham Mill	1.114	0.223	0.095	0.053
34014	Wensum at Swanton Morley Total	1.109	0.220	0.094	0.052
35008	Gipping at Stowmarket	1.104	0.218	0.093	0.051
36005	Brett at Hadleigh	1.112	0.222	0.095	0.053
36007	Belchamp Brook at Bardfield Bridge	1.100	0.216	0.092	0.051
37001	Roding at Redbridge	1.109	0.221	0.094	0.052
37011	Chelmer at Churchend	1.080	0.207	0.087	0.048
37019	Beam at Bretons Farm	1.070	0.202	0.084	0.046
38003	Mimram at Panshanger Park	1.080	0.207	0.087	0.048
38014	Salmon Brook at Edmonton	1.079	0.206	0.086	0.047
39001	Thames at Kingston	1.094	0.213	0.090	0.050
39006	Windrush at Newbridge	1.036	0.187	0.076	0.041
39034	Evenlode at Cassington Mill	1.045	0.191	0.078	0.042
39049	Silk Stream at Colindeep Lane	1.066	0.200	0.083	0.045
39057	Crane at Cranford Park	1.073	0.203	0.085	0.046
39076	Windrush at Worsham	1.002	0.173	0.068	0.036
39081	Ock at Abingdon	1.076	0.205	0.086	0.047
39090	Cole at Inglesham	1.040	0.189	0.077	0.041
39096	Wealdstone Brook at Wembley	1.081	0.207	0.087	0.048
39105	Thame at Wheatley	1.054	0.195	0.080	0.043
39131	Brent at Costons Lane Greenford	1.085	0.209	0.088	0.048
40003	Medway at Teston	1.113	0.222	0.095	0.053
40011	Great Stour at Horton	1.095	0.214	0.090	0.050
40017	Dudwell at Burwash	1.018	0.179	0.072	0.038
40023	East Stour at South Willesborough	1.040	0.189	0.077	0.041
41011	Rother at Iping Mill	0.894	0.131	0.048	0.024
41022	Lod at Halfway Bridge	0.964	0.157	0.061	0.031
41026	Cockhaise Brook at Holywell	0.947	0.151	0.057	0.029
42012	Anton at Fullerton	1.079	0.206	0.086	0.047
43003	Avon at East Mills	0.817	0.106	0.036	0.017
43005	Avon at Amesbury	1.050	0.193	0.079	0.043
43006	Nadder at Wilton	1.045	0.191	0.078	0.042
43007	Stour at Throop	1.105	0.218	0.093	0.052
43021	Avon at Knapp Mill	1.106	0.219	0.093	0.052
44002	Piddle at Baggs Mill	1.105	0.218	0.093	0.052
45001	Exe at Thorverton	0.897	0.132	0.048	0.024
45004	Axe at Whitford	1.042	0.190	0.077	0.042

(continued on next page)

Table 3 (*continued*)

Station		WFD P status boundary value (mg/L)			
Station	Name	Poor	Moderate	Good	High
45005	Otter at Dotton	1.024	0.182	0.073	0.039
47001	Tamar at Gunnislake	0.896	0.132	0.048	0.024
47008	Thrushel at Tinhay	0.848	0.116	0.040	0.020
47014	Walkham at Horrabridge	0.759	0.088	0.028	0.013
48003	Fal at Tregony	0.833	0.111	0.038	0.018
49001	Camel at Denby	0.882	0.127	0.046	0.023
50002	Torridge at Torrington	0.878	0.126	0.045	0.022
50006	Mole at Woodleigh	0.874	0.124	0.045	0.022
50007	Taw at Taw Bridge	0.844	0.114	0.040	0.019
51001	Doniford Stream at Swill Bridge	1.055	0.195	0.080	0.044
52010	Brue at Lovington	1.106	0.219	0.093	0.052
53005	Midford Brook at Midford	1.097	0.215	0.091	0.050
53006	Frome (Bristol) at Frenchay	1.077	0.205	0.086	0.047
53017	Boyd at Bitton	1.119	0.225	0.097	0.054
53018	Avon at Bathford	1.093	0.213	0.090	0.050
54001	Severn at Bewdley	1.010	0.176	0.070	0.037
54008	Teme at Tenbury	0.994	0.169	0.067	0.035
54036	Isbourne at Hinton on the Green	1.076	0.205	0.086	0.047
54038	Tanat at Llanyblodwel	1.018	0.179	0.072	0.038
54057	Severn at Haw Bridge	1.049	0.193	0.079	0.043
55002	Wye at Belmont	1.062	0.198	0.082	0.045
55003	Lugg at Lugwardine	1.080	0.207	0.087	0.048
68001	Weaver at Ashbrook	1.039	0.188	0.077	0.041
68005	Weaver at Audlem	1.055	0.195	0.080	0.044
71001	Ribble at Samlesbury	1.030	0.185	0.075	0.040
71006	Ribble at Henthorn	1.022	0.181	0.073	0.039
71009	Ribble at New Jumbles Rock	0.996	0.170	0.067	0.035
72004	Lune at Caton	0.861	0.120	0.042	0.021
72014	Conder at Galgate	0.967	0.159	0.061	0.032
72015	Lune at Lunes Bridge	0.848	0.115	0.040	0.020
73005	Kent at Sedgwick	0.955	0.154	0.059	0.030
73009	Sprint at Sprint Mill	0.825	0.108	0.037	0.018
73011	Mint at Mint Bridge	0.923	0.141	0.053	0.027
73013	Rothay at Miller Bridge House	0.786	0.096	0.032	0.015
73014	Brathay at Jeffy Knotts	0.786	0.096	0.032	0.015
74001	Duddon at Duddon Hall	0.794	0.099	0.033	0.015
74005	Ehen at Braystones	0.931	0.145	0.054	0.028
74007	Esk at Cropple How	0.775	0.093	0.030	0.014
75017	Ellen at Bullgill	0.988	0.167	0.065	0.034
76005	Eden at Temple Sowerby	0.979	0.163	0.063	0.033
76007	Eden at Sheepmount	1.013	0.177	0.071	0.038
76008	Irthing at Greenholme	0.989	0.167	0.066	0.034

Van Vliet and Zwolsman, 2008), due to increased residence times and slow flowing rivers (Johnson et al., 2009). However, the role of droughts and subsequent runoff in delivering greater amounts of diffuse nutrients under a more variable climate has not been quantified. Furthermore, some climate studies indicate increases in the magnitude and frequency of short droughts (<18 months) in future, but there is little information on changes in longer droughts (Watts and Anderson, 2013; Watts et al., 2015). In France, during dry years, eutrophication by phytoplankton and spring algal blooms threaten drinking water supply, even after reducing point sources by 85%, leading to the conclusion that wastewater treatment must be accompanied by measures to reduce diffuse sources linked to agricultural inputs (Garnier et al., 2005).

Nationally, there is a clear trend of increasing TRP concentration in future, especially during the summer. However, the relatively small changes make it hard to understand the implications for both WFD status and biological response. In addition, changes in water temperature could lead to earlier onset of algal blooms (Bowes et al., 2016) or changes in the occurrence of important thermal thresholds so that these results, whilst indicative, should be used with caution.

Other sources of uncertainty in the projected estimates of phosphorus concentrations come from the relationship between flow and P in the current day; uncertainty in future projections of flow and understanding of P dynamics. Sources of error associated with the paired TRP and river flow data (including co-location as well as record length issues), and associated with fitting regression relationships to empirical data (especially where there is limited observational data of high flows where much of the load may be shifted) were discussed in the methods section of this paper. There is an implicit assumption that the model parameters remain valid in the future which may not be correct. This may become more of an issue under changing land use patterns and population growth; the relative importance of all three pressures needs to be explored to derive a more complete picture of risk than provided by this initial climate screening approach. Furthermore, although rainfall and potential evaporation are taken into account in the future river flow estimates and the LAM implicitly considers the combined impact of all sources by focusing on loads, this analysis doesn't explicitly consider the impact of changing rainfall patterns on delivery to the river. The seasonal and spatial variation of rainfall and potential evaporation and its interaction with catchment characteristics has been shown to have a strong influence on future river flows (e.g. Charlton and Arnell, 2014). Other studies (e.g. De Paola et al., 2014) have shown the influence of climate change on altering the intensity, duration and frequency of rainfall curves. Such changes may not only influence river flow directly but can indirectly affect stormwater outflows (e.g. De Paola and Ranucci, 2012) and sediment delivery characteristics. Increased sediment delivery during heavy rainfall, may increase P concentrations during such events (e.g. Ockenden et al., 2016), altering the pattern of eutrophication risk. It remains to be seen whether efforts to control losses of nutrients from land into rivers by changing land management practices will be able to combat the effects of potentially increased P delivery from changes in runoff. Despite this, the results here suggest that P concentrations will remain high enough to fail P standards into the

Fig. 6. (a) Maximum, median, and minimum maps of baseline WFD status. (b) Maximum, median, and minimum maps of 2050s WFD status.

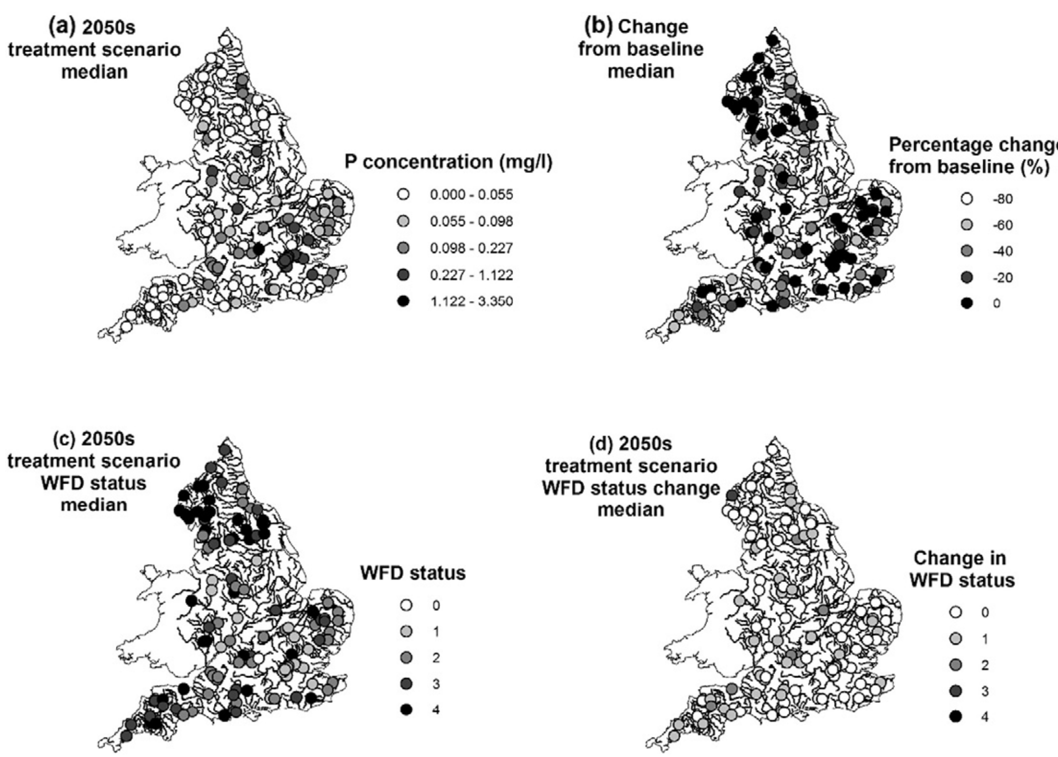

Fig. 7. Median annual average under treatment scenario: (a) absolute 2050s P concentration (mg/L). (b) percentage change between 2050s and baseline. (c) 2050s WFD status. (d) Change in WFD status.

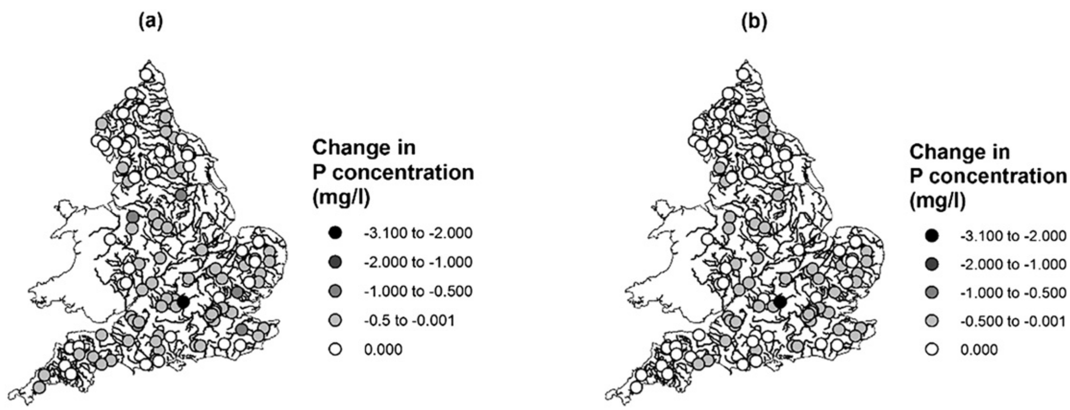

Fig. 8. Additional reductions needed to achieve good status for median annual average (2050s): (a) from original P projections. (b) from treatment scenario.

future and that other drivers of eutrophication risk need to be investigated in order to understand future risk.

There is greater variability in future P concentration between sites than between climate scenarios. However, the climate ensemble variability that is introduced can make interpretation of the broader patterns of median and range of change challenging. Uncertainty in the flow ensemble originates from the climate projections themselves and the hydrological modelling conducted using the climate information. Both of these aspects have been explored in detail elsewhere (see Prudhomme et al., 2012a, 2013).

Uncertainty in the LAM and flow ensemble interact. There is a possibility that change in P concentration may be underestimated if the flow/P concentration relationships significantly misrepresent baseline conditions. This may have occurred as a result of how we derived P/flow relationships at some sites. For a point source dominated site, if FFH underestimates low flows, concentrations will be overestimated (and vice-versa).

There are also some uncertainties in the future treatment scenario used. The value of the LAM *A* parameter (P from point sources) should decrease in the future treatment scenario. This was not always the case because: (1) the value is based on consented discharges served by the works where the actual P may not be known; (2) the calculated discharge volume of 180 L/person/day is an estimate, and will in-practice vary between catchments; (3) the A parameter includes *other* constant inputs such as P from groundwater and septic tank misconnections and (4) the value of *A* will depend on the level of sewage treatment already present in the catchment. If the majority of large works in a river waterbody have already implemented tertiary treatment then the additional reduction in point source load that can be achieved through better treatment will be small.

Our treatment scenario to reduce STWs discharge to <0.5 mg/L of P is technically feasible for most sites and does result in some sites meeting regulatory standards but in general is insufficient to change the current patterns of failure. The poor status boundaries for WFD P standards are much wider than the envelope for good status boundaries. This means that not changing status boundaries doesn't necessarily mean that there is no significant effect on ecosystems. Annual average changes in P concentration are quite insensitive to flow changes but these are used to determine WFD status boundaries. In reality 'typical summer' concentrations may be a more important indicator of ecological condition (Bowes et al., 2014; Jarvie et al., 2006), and specific 'low flow and warm periods of longer duration' P concentrations may be a more significant indicator of risks of excessive algal growth.

Currently the main management approach to eutrophication is to reduce nutrient inputs to rivers, in particular through sewage treatment. This may be harder to achieve in future with increasing population and agricultural intensification (Johnson et al., 2009) and especially if climate change increases P concentrations by altering flow regimes as indicated in this analysis. Furthermore, improvements through additional wastewater treatment are of limited benefit where sites are dominated by diffuse sources of P, either because wastewater treatment is at its limit of effectiveness or because diffuse sources truly dominate. In such cases, improving management of these diffuse sources is necessary to improve ecological status. Meeting ecological objectives might be attained more cost-effectively by controlling light conditions through more riparian shading (Hutchins et al., 2010; Bowes et al., 2012b). A better understanding of the link between P standards and algal growth might help target effective interventions. However, to understand eutrophication risk requires understanding the other drivers of that risk. These may include understanding the seasonal circumstances that lead to eutrophication (i.e. temporal dynamics of P, sediment P retention, flow, light and temperature) (Bowes et al., 2016) that might be hidden by generalised flow-P models. This is especially important given that estimated P concentrations are frequently high enough to meet the threshold for algal growth: other factors need to be investigated to understand eutrophication risk more fully.

We have high confidence in climate projections for increasing air temperature but less confidence in future patterns of rainfall and river flow. In addition, climate and eutrophication impacts on ecosystems are even harder to determine. For example, heavier or more frequent winter rainfall could increase nutrient loads derived from land (Antunes and Rodrigues, 2011). The aquatic microbial community is likely to respond to changes in flow regime in different ways; for example, phytoplankton biomass may be more sensitive to changes in flow rate, light and water temperature than to nutrients (Hutchins, 2012). These other factors will need to be considered in a more comprehensive assessment of eutrophication risk in rivers along with residence times.

5. Conclusions

We've used existing information on future river flows to project future phosphorus concentrations. This novel national scale assessment of a climate change impact on P concentration status directly informs strategic decision making at a national scale by improving understanding of when and where action needs to be taken to prevent deterioration in water quality. Our future maps of TRP concentration take account of climate change impacts on flow and are a useful first step in understanding the future risk of eutrophication in English rivers.

Predicted climate change impacts on flow tend to result in small but inconsistent increases in annual average P concentrations in rivers and greater increases particularly in summer, when the risk of excessive algal growth is highest. Importantly, most P concentration estimates are sufficiently high for meeting thresholds for algal growth suggesting the need for further management and to understand other drivers of eutrophication risk such as residence times, water temperature and sunlight duration.

Introducing P stripping into more water treatment plants can help increase WFD status. However, currently planned P management

interventions are inadequate to meet WFD objectives and these results indicate how much more would need to be done to meet existing P standards if rivers flows change as currently projected.

The scenarios within this study specifically look at flow change effects on P concentration, and future inputs from land use activities or population changes were not investigated. These results suggest that incorporating a change factor for future P estimates is needed alongside estimates of P delivery related to these other pressures. Even so, climate change impacts on river flow do lead to some sites dropping a WFD status band. The changes in WFD status boundaries are relatively small (mostly deteriorating in ecological condition). However this work is the first step in exploring future eutrophication risk, and changes in other factors, such as temperature and flow, may be more significant. These factors need to be considered alongside the biological response to the flow driven increases in P concentrations.

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